Innovative uses of vegetated drainage ditches for reducing agricultural runoff

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Abstract Vegetated agricultural ditches play an important role in mitigation of pesticides following irrigation and storm runoff events. In a simulated runoff event in the Mississippi (USA) Delta, the mitigation capacity of a drainage ditch using the pyrethroid esfenvalerate (Asana XL™) was evaluated. The pesticide was amended to soil prior to the runoff event to simulate actual runoff, ensuring the presence of esfenvalerate in both water and suspended particulate phases. Water, sediment, and plant samples were collected temporally and spatially along the drainage ditch. Even with mixing of the pesticide with soil before application, approximately 99% of measured esfenvalerate was associated with ditch vegetation (*Ludwigia peploides*, *Polygonum amphibium*, and *Leersia oryzoides*) three hours following event initiation. This trend continued for the 112 d study duration. Simple modeling results also suggest that aqueous concentrations of esfenvalerate could be mitigated to 0.1% of the initial exposure concentration within 510 m of a vegetated ditch. Observed field half-lives in water, sediment, and plant were 0.12 d, 9 d, and 1.3 d, respectively. These results validate the role vegetation plays in the mitigation of pesticides, and that ditches are an indispensable component of the agricultural production landscape.

Keywords BMP; ditch; drainage; esfenvalerate; plants

Introduction

Due to growing water quality concerns, agriculture has placed an increased emphasis on discovering new, innovative best management practices (BMPs) to decrease the effects of storm water runoff containing nutrients, bacteria, sediment, and pesticides. Concerted efforts between the United States Department of Agriculture Natural Resource Conservation Service (USDA NRCS), USDA Agricultural Research Service (USDA ARS), and the National Association of Conservation Districts (NACD) have led to the development of acceptable BMPs such as stiff grass hedges, constructed wetlands, conservation tillage, slotted-board risers, and winter cover crops to decrease the amount of potential agricultural contaminants leaving fields. Many of these BMPs have either one-time or continual costs involved, such as planting and maintenance of hedges, procurement of drop-tiles for risers, or removal of production acreage for implementation of constructed wetlands. Current research suggests using vegetated agricultural drainage ditches as an additional BMP for pesticide mitigation. No construction costs should be incurred, with the exception of only marginal, occasional maintenance to prevent serious flow constriction. Research into the use of agricultural drainage ditches as buffers for agricultural storm water runoff has been scarce until recent years.

Much debate has arisen over agricultural drainage ditches, perhaps due to differences in operational definitions. For the purpose of this research, drainage ditches refer to those structures, already within the agricultural production landscape, used as a conduit for water between the field and a receiving water body. Drainage ditches are often thought of solely as a means for water transport from production acreage to an aquatic receiving system; however, their environmental benefits (transfer and transformation of contaminants) have often been overlooked. Few studies have addressed benefits of vegetated drainage ditches in decreasing concentrations of potential agricultural contaminants (Moore *et al.*, 2001; Crum *et al.*, 1998; Meuleman and Beltman, 1993; Drent and Kersting, 1992).

Esfenvalerate [(S)-alpha-cyano-3-phenoxybenzyl(S)-2-(-4-chlorophenyl)-3-methylbutyrate] is a fourth generation pyrethroid insecticide sold under the trade name Asana XL™. Approximately 104,000 kg of esfenvalerate (as active ingredient) was applied on 49 different US crops covering over 1.5 million hectares of land in 1997 (NCFAP, 2000). Objectives of the current research include 1) evaluation of esfenvalerate partitioning (water, sediment, and plant) within an agricultural drainage ditch and 2) determine the necessary ditch length for effective mitigation of esfenvalerate, given recommended field application rates and other underlying assumptions regarding rainfall and runoff variability.

Materials and methods

Ditch exposure

A 600 m agricultural drainage ditch located immediately west of Thighman Lake, Sunflower County, Mississippi, USA, was used to determine the mitigation capacity of such systems for esfenvalerate-associated runoff. The ditch is located within the Mississippi Delta Management Systems Evaluation Area (MDMSEA), which is a consortium of Federal, state, and local agencies, as well as industry and participating farmers. Structural measurements were collected on Thighman ditch prior to esfenvalerate exposures. Top bank width was 4.5 m, with an average water width of 2.8 ± 0.06 m, and mean water depths of 0.27 ± 0.04 m (left bank), 0.31 ± 0.03 m (mid-ditch), and 0.27 ± 0.04 m (right bank). Ditch current velocity prior to event simulation was 0.02 ± 0.01 m/s. Sampling sites for the study were established at the runoff injection point (0 m), 10, 20, 40, 80, 100, 200, 400, and 600 m below the injection point. Twenty-four hours prior to event simulation, percent plant cover and water quality parameters were collected throughout the ditch for reference information (Table 1). Replicate ditch transect sampling was accomplished using a 0.28 m^2 sample quadrat.

A simulated storm runoff event was conducted on the designated ditch on August 2, 2000. A mixture of esfenvalerate (as Asana XL™), water, and suspended sediment (400 mg/L) was amended into the ditch to simulate a storm runoff event. Esfenvalerate concentrations (0.15 mg/L) were based on recommended application rates (0.035 kg active ingredient per ha) and an assumed 1% pesticide and 1% water runoff from a 0.64 cm storm event across a 20 ha contributing area. The simulated runoff event was accomplished using a 2.0 m length of 7.6 cm diameter PVC pipe with 16, 1.5-cm holes evenly dispersed along the pipe's length for diffusion. Four, 3,800 L water tanks filled with groundwater were connected (one at a time) to the diffuser and used as a water source for the simulated event. Sediment was amended with esfenvalerate less than 24 h prior to initiation of the simulated event. Once on site, the esfenvalerate-sediment amendment was added to water in a 110 L mixing chamber, then delivered to the top of the PVC diffuser via Tygon™ tubing using an Atwood™ V450 submersible pump at a rate of 0.02 L/s for 90 minutes. A 5-cm hose delivered water from the 3,800-L water tanks directly to the diffuser at a rate of approximately 1 L/s.

Collection of water, sediment, and plant samples

One liter amber bottles were used to collect grab samples of water at 24 h prior to exposure, beginning of exposure (0 h), every 15 minutes for 3 h, then at intervals of 6 h, 12 h, 24 h, 48 h, 7 d, 14 d, 28 d, and 42 d post-application from each site. After collection, water samples were immediately extracted in the field. Remaining samples were placed on ice and transported back to the laboratory for final extraction. Sediment and plant samples were collected at –24 h, 3 h, 12 h, 24 h, 7 d, 14 d, 28 d, 42 d, and 112 d post-application, wrapped in aluminium foil and immediately placed on ice for transport back to the laboratory. Sediment and plant samples were frozen (–10°C) until being dried for analysis. Sediment samples were collected from the top 3 cm using sterilized scoops, while plant material was obtained using acetone-rinsed scissors. Only that plant material exposed in the water column was collected for analysis. (Disparity between the final sediment and plant collections and water collections are the result of the ditch remaining dry for an extended period of time following the 42 d sample collection.)

Extraction and analysis of water, sediment and plant samples

Water, sediment and plant samples were extracted using previous methods described by Bennett *et al.* (2000) and Moore *et al.* (2001). Briefly, water samples were extracted by liquid–liquid extraction using ethyl acetate, while both sediment and plant samples were extracted by ultrasonication using ethyl acetate. Sediment and plant extracts were also subject to silica gel cleanup before analysis.

Esfenvalerate was analyzed by gas chromatography-microelectron capture detection (GC-μECD) using a HP 6890 gas chromatograph equipped with a 30 m DB-1MS column. The following oven temperature program was used: 75°C (held for 1 min) to 225°C at a rate of 40°C/min. The injector and detector temperatures were set to 250°C and 325°C, respectively. The carrier gas, ultra high purity helium (nexAir, Memphis, TN), was set to a constant flow of 1 mL/min and makeup gas, ultra high purity nitrogen (nexAir, Memphis, TN), was set at a constant makeup flow of 60.0 mL/min. A multi-level calibration procedure was used with standards and was updated every ninth sample. Limits of detection (LOD) for esfenvalerate in water, sediments and plants were 0.01 ng/mL, 0.01 ng/g, and 0.01 ng/g, respectively. Mean extraction efficiencies, based on fortified samples, were > 90% for water, sediment, and plants.

Modeling of pesticide transport

Ordinary least-square linear regression analyses were used to fit curves to base 10 log-transformed esfenvalerate water, sediment, and plant tissue concentrations (y) versus log-transformed distance down ditch from the injection point (x). For simplicity, only the maximum concentrations observed at each sampling site (regardless of time) were used in the analyses. In order to derive regression formulas to predict the distance required to achieve a given concentration, similar regressions were also performed using log-transformed concentration as the independent (x) variable and distance as the log-transformed dependent (y) variable. More complex nonlinear functions were also fit to the data, but produced only marginal improvements in fit.

Table 1 Water quality parameters of Thighman Ditch, Mississippi, USA, August 2000 prior to esfenvalerate exposure

Parameter	Units	Value
Velocity	m/s	0.01 ± 0.003
Dissolved oxygen	mg/L	2.70 ± 1.8
Temperature	°C	27.7 ± 0.2
Conductivity	μS/cm	460 ± 4
pH	s.u.	7.0 ± 0.1

Results and discussion

Esfenvalerate fate

Three hours following initiation of the simulated storm runoff, 98 ± 1% of the total measured esfenvalerate (across the entire ditch) was associated with plant material, while less than 1% was associated with the water column. The importance of vegetation was evident at the injection point (0 m), where at 3 h, 79% of the measured esfenvalerate (97 μg/L) was associated with the water column, while 21% (25.5 μg/kg) was associated with sediment. By the 12 h sample, 100% of the measured esfenvalerate was associated with sediment (165 μg/kg). Excluding the injection site, 86 ± 8% of the measured esfenvalerate in the entire ditch was associated with plant material, while 14 ± 8% was associated with the sediment at 12 h.

For all sampling locations and times, mean esfenvalerate concentrations were 4 ± 2 μg/L, 17 ± 12 μg/kg, and 784 ± 426 μg/kg for water, sediment, and plant material, respectively. Evaluating each sampling location independently, 54% (4,170 ppb – combining water [μg/L], sediment [μg/kg], and plant material [μg/kg]) of the measured esfenvalerate during the study duration was measured at the 10 m sampling location. Fifteen percent and 10% of the measured esfenvalerate (1,139 ppb and 751 ppb) were associated with the 40 m and 20 m sites, respectively. Observed field half-lives of esfenvalerate in the ditch were 0.12 d, 9 d, and 1.3 d for water, sediment, and plant, respectively.

Modeling

Maximum observed esfenvalerate concentrations were inversely proportional to distance downstream from the point of injection, producing regression coefficients that were significant at $p < 0.001$ (Figure 1). Standard errors ranged from 0.4 to 0.5 (in terms of log-transformed concentration), but 95% confidence intervals were rather broad (spanning about five orders of magnitude in terms of untransformed concentration) due to the relatively small number of data points. The formulas predict that a ditch length of 500 m would have been adequate to reduce concentrations of esfenvalerate to concentrations of 0.3 μg/L, 3.4 μg/kg, and 13 μg/kg, in water, sediment, and plants, respectively. These levels represent about 0.2%, 0.5% and 0.1% of the maximum observed concentration at the injection point. When similar regressions were run using the same log-transformed data but with distance as the dependent variable, measures of goodness-of-fit were similar. The resulting formulas predicted that ditch lengths of 509, 423, and 406 m would be required to reduce maximum esfenvalerate concentrations in water, sediment, and plants, respectively, to 0.1% of the maximum value at the injection point (Table 2).

Little argument exists over the historical impact agriculture has played on surface water quality. Agricultural production uses approximately 35% of accessible freshwater runoff (Vitousek *et al.*, 1997; Robertson, 2000). Before implementation of BMPs and awareness of positive qualities of wetlands, agriculture had great difficulty with erosion and other storm runoff complications. However, since the implementation of the Clean Water Act and other environmentally driven initiatives, agricultural research assessing threats to water quality has centred on various areas of inquiry, with one being primarily the range of BMPs for non-point source pollution reduction (Watson *et al.*, 1994; Watzin and McIntosh, 1999).

Drainage ditch research

While several BMPs for agricultural runoff reduction are available for farmers and conservationists to use, one often-overlooked BMP is the agricultural drainage ditch. Initial research with drainage ditches has primarily been limited to work done in The Netherlands. Drent and Kersting (1992) reported on the importance of drainage ditches, since 300,000

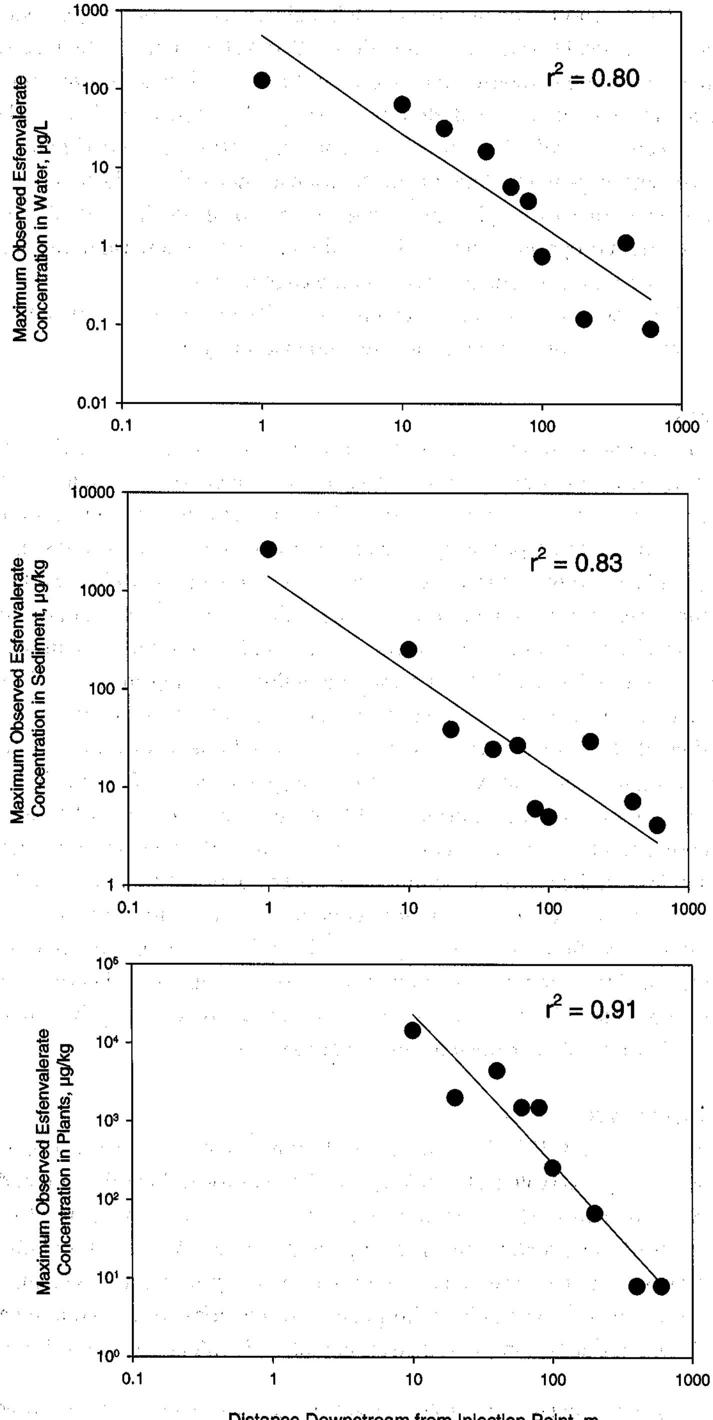

Figure 1 Least-squares regression relationships fit to log-transformed maximum observed esfenvalerate concentration versus distance downstream from injection point. Injection point concentrations in water and sediment are plotted at $x = 1$ on log scale

km of the important aquatic ecosystems existed in The Netherlands. A majority of the research has focused on the ditch banks themselves, which may serve as refugia for endangered plant species, as well as other ditch management issues (Van Strien *et al.*, 1989, 1991). Janse (1998) reported on the annual ditch maintenance involving removal of vegetation and detrital matter. This was in conjunction with development of a ditch eutrophication model. Other research has focused on the use of ditches and reed marshes for successful nutrient reduction (Meuleman and Beltman, 1993). De Snoo and Van der Poll (1999) focused on effects of herbicide drift on boundary vegetation and noted that many Dutch farmers consistently sprayed edges of the fields with herbicides, thereby increasing the probability of drift down the ditch bank. Earlier, De Snoo and de Wit (1998) reported a 3 m buffer between field crops and ditch banks resulted in a 95% decrease in pesticide deposition on ditch banks and subsequently, within ditches themselves.

The fate of the herbicide linuron was examined by Crum *et al.* (1998) in outdoor ditches, and they reported a high linuron sorption rate onto macrophytes.

Conclusions

Research in the Netherlands led US researchers to previous (Moore *et al.*, 2001) and current efforts to use agricultural drainage ditches to decrease concentrations of agricultural pesticides associated with storm runoff. Much debate has arisen over agricultural drainage ditches in the United States, primarily focusing on maintenance versus construction issues (Grumbles, 1991). Current research promotes the use of historical drainage ditches already present in the agricultural production landscape. Initial studies using ditches to decrease concentrations of the herbicide atrazine and pyrethroid lambda-cyhalothrin were successful, with 61% of atrazine and 87% of lambda-cyhalothrin being measured in ditch plant material one hour following a simulated storm runoff event (Moore *et al.*, 2001). Erstfield (1999) evaluated the fate of synthetic pyrethroids deltamethrin and tralomethrin and reported their significant affinity to accumulate in plant tissue. While it is important to emphasize the use of agricultural drainage ditches as an alternative BMP for agricultural pesticide runoff, additional emphasis should be placed on the vegetation within the ditch, which is often less studied than the sediment or water fractions. According to Cousins and Mackay (2001), models analyzing organic contaminant fate rarely possess a vegetation component since it is often assumed that partitioning is limited primarily to water and sediment. As evidenced in current and previous ditch studies, vegetation plays a significant role in the sorption of pesticides from the water column, thereby reducing the threat of downstream contamination and detriment to aquatic organisms.

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Table 2 Summary of linear regression formulas produced using distance downstream from injection point versus maximum esfenvalerate concentrations. Data were log-transformed. Relations are power functions of the form $y = a\,(x+1)^b$ where y is the distance in metres and x is concentration in µg/L (water) or µg/kg (sediment, plants) divided by maximum observed concentration at the injection point

Media	No. of observations	*a*	*b*	r^2	*P*
Water	10	3.75	−1.20	0.80	0.0005
Sediment	10	1.89	−0.97	0.83	0.0002
Plants	9	186.9	−1.97	0.91	0.00006

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References

Bennett, E.R., Moore, M.T., Cooper, C.M. and Smith, Jr., S. (2000). Method for the simultaneous extraction and analysis of two current use pesticides, atrazine and lambda-cyhalothrin in sediment and aquatic plants. *Bull. Environ. Contam. Toxicol.*, **64**, 825–833.

Cousins, I.T. and Mackay, D. (2001). Strategies for including vegetation compartments in multimedia models. *Chemosphere*, **44**(4), 643–654.

Crum, S.J.H., Aalderink, G.H. and Brock, T.C.M. (1998). Fate of the herbicide linuron in outdoor experimental ditches. *Chemosphere*, **36**(10), 2175–2190.

De Snoo, G.R. and de Wit, P.J. (1998). Buffer zones for reducing pesticide drift to ditches and risks to aquatic organisms. *Ecotoxicol. Env. Saf.*, **41**, 112–118.

De Snoo, G.R. and Van der Poll, R.J. (1999). Effect of herbicide drift on adjacent boundary vegetation. *Ag. Ecosys. Environ.*, **73**(1), 1–6.

Drent, J. and Kersting, K. (1992). *Experimental ditches for ecotoxicological experiments and eutrophication research under natural conditions: A technical survey*. Report 65. DLO Winand Staring Centre, Wageningen, The Netherlands.

Erstfield, K.M. (1999). Environmental fate of synthetic pyrethroids during spraydrift and field runoff treatments in aquatic microcosms. *Chemosphere*, **39**(10), 1737–1769.

Grumbles, B. (1991). Wetlands, drainage ditches, and the Clean Water Act. *J. Soil Wat. Conserv.*, **46**(3), 174–177.

Janse, J. (1998). A model of ditch vegetation in relation to eutrophication. *Wat. Sci. Tech.*, **37**(3), 139–149.

Meuleman, A.F.M. and Beltman, B. (1993). The use of vegetated ditches for water quality improvement. *Hydrobiologia*, **253**(1–3), 375.

Moore, M.T., Bennett, E.R., Cooper, C.M., Smith, S. Jr., Shields, F.D., Milam, C.D. and Farris, J.L. (2001). Transport and fate of atrazine and lambda-cyhalothrin in an agricultural drainage ditch in the Mississippi Delta, USA. *Ag. Ecosys. Environ.*, **87**, 309–314.

National Center for Food and Agricultural Policy (NCFAP) (2000). National pesticide use database, Washington, D.C. *http://www.ncfap.org*

Robertson, A.I. (2000). The gaps between ecosystem ecology and industrial agriculture. *Ecosys.*, **3**, 413–418.

Van Strien, A.J., Van Der Linden, J., Melman, T.C.P. and Noordevliet, M.A.W. (1989). Factors affecting the vegetation of ditch banks in peat areas in the western Netherlands. *J. App. Ecol.*, **26**, 989–1004.

Van Strien, A.J., Van Der Burg, T., Rip, W.J. and Strucker, R.C.W. (1991). Effects of mechanical ditch management of the vegetation of ditch banks in Dutch peat areas. *J. App. Ecol.*, **28**, 501–513.

Vitousek, P.M., Mooney, H.A., Lubchenco, J. and Melillo, J.M. (1997). Human domination of earth's ecosystems. *Science*, **277**, 494–499.

Watson, J., Hassinger, E., Reffruschinni, K., Sheedy, M. and Anthony, B. (1994). Best management practices meeting water quality goals. *J. Soil Wat. Conserv.*, **49**(2), 39–42.

Watzin, M.C. and McIntosh, A.W. (1999). Aquatic ecosystems in agricultural landscapes: A review of ecological indicators and achievable ecological outcomes. *J. Soil and Wat. Conserv.*, **54**(4), 636–644.